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Thinning and Prescribed Fire Effects on Fuels and Potential Fire Behavior in an Eastern Cascades Forest, Washington, USA

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Abstract

Prescribed fire and low thinning were applied to dry forests dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) in the eastern Washington Cascades. Experimental design was an unbalanced analysis of variance with 4 control units, 4 thin units, 2 burn units and 2 thin/burn units. Thinning was designed to reduce basal area to 10–14 m^2 ha^{-1} in a non-uniform pattern and burning was a low intensity spring burn. Burn coverage was spotty, ranging from 23–51%, and considered ineffective in reducing fuels at the time of application by management and research personnel. Both thinning and burning had effects on vegetation and fuels variables. Thinning reduced canopy closure, canopy bulk density, and basal area, and increased canopy base height. Burning had no influence on these canopy variables. Thinning increased 10-hr timelag (0.62–2.54 cm) fuels. Burning decreased 1-hr (0–0.62 cm) and 10-hr timelag fuels, forest floor depth and mass, and increased fuelbed depth. There were interactions between thinning and burning for 1-hr and 10-hr timelag fuels, and fuelbed depth. These differences in fuel properties did not translate into differences in simulated wildfire behavior and tree mortality. Thinning did

increase potential surface fire flame length under 97 percentile weather, and active crown fire potential decreased on thinned units, but basal area survival did not significantly differ between treatments under 80 and 97 percentile weather. The scale at which data are presented has a large influence on interpretation of results. For example, torching fire behavior, expressed as an average at the unit level, was low, but 17% of the individual plots (about 30 plots total per unit) across all treatments did exhibit potential torching behavior.

Keywords

prescribed fire thinning fuel reduction Washington Cascades vegetation effects *Pinus ponderosa Pseudotsuga menziesii*

Introduction

Fire exclusion policies in the 20th century created unnaturally high fuel accumulations of dead and live fuels in many dry Western forests that once experienced low-moderate severity fire regimes (Allen et al. 2002, Brown et al. 2004). Area burned by wildland fire has significantly increased in the last 20 years, and although part of the extent issue (e.g., fire size) can be explained by climate (Westerling et al. 2002, Westerling et al. 2006), increased severity (e.g., tree mortality) from these fires is more likely linked to accumulated fuels (Agee 1997, Fitzgerald 2002). The link between historical fire size and drought in dry forests is well established (Heyerdahl et al. 2002), but stand reconstructions in these forests show that many trees survived 20–30 fires, inferring a low-severity fire regime historically was present (Wright and Agee 2004). Programs to deal with the problem of larger and more severe fires in the West have been initiated at the state level (Western Governor's Association 2003: Policy Resolution 03–18, September 15, 2003, at Big Sky, Montana) and the Federal level (Healthy Forests Restoration Act of 2003). Focus has been both on values (the wildland-urban interface) and hazard (fuel reduction), including timing (Allen et al. 2002) and treatment type (Agee and Skinner 2005).

The environmental effects of restoration treatments are broadly known, but much remains to be learned. In 1999, a consortium of Federal, non-governmental organizations, and university scientists successfully competed through the Federal Joint Fire Science Program for a national network of study sites (now totaling 13) where effects of restorative treatments including fire and fire surrogates such as thinning could be studied using comparable methods. Variables at each study site included aspects of plant community changes, fuels, soils, birds and small mammals, insects and disease (http://www.ffs.fs.fed.us). One of the Pacific Northwest study sites was Mission Creek on the Okanogan-Wenatchee National Forests in Washington State.

In this paper, we present the results of the fuels and fire behavior portion of the project. Our null hypotheses were that thinning and burning would have no effect on stand structure relevant to fire behavior, fuels, or fire behavior and effects.

Study Site

The Mission Creek study site (Figure 1) is located in the Wenatchee Mountains east of the crest of the Cascade Mountains in Washington (roughly 47°25′50″N, Longitude 120°32′55″W). The climate is a mix of maritime and continental climates due to proximity to low mountain passes along the Cascade crest. Regional winds are primarily westerly, and orographic effects result in the study area being much drier than areas west of the Cascade crest. Vegetation is typical of dry forests of eastern Washington. Historically, most low-elevation forests in the area were dominated by ponderosa pine (*Pinus ponderosa*) with some Douglas-fir (*Pseudotsuga menziesii*) and occasionally grand fir (*Abies grandis*) and western larch (*Larix occidentalis*) (Agee 1994, Harrod et al. 1999). Reconstructions of forest basal area and density for these forests range from 10–20 m^2 ha^{-1} and 30–70 trees ha^{-1} (Harrod et al. 1999, Agee 2003). Potential vegetation consists of dry to mesic Douglas-fir plant association groups, but frequent disturbance by fire (Everett et al. 2000, Wright and Agee 2004) prevented much late successional forest development on these sites. On similar forest plant associations in the area, Everett et al. (2000) found mean fire return intervals of 6–8 years using a composite of all fire scars over 580–650 ha areas. Wright and Agee (2004) used a compositing technique over much smaller areas (0.5–2.5 ha) and found mean fire return intervals of 18.8-20.6 years. Over 80 percent of the fires identified were “late season” fires, occurring in the latewood of the scarred annual ring or after growth for the year had ceased. Due to the fire resistance of the major tree species, large trees in stands with a single canopy layer dominated these landscapes in a classic low-severity fire regime (Agee 1993, Agee 2003, Wright and Agee 2004). Stands were often clumpy (Harrod et al. 1999), with substantial herbaceous cover in open areas (Figure 2).

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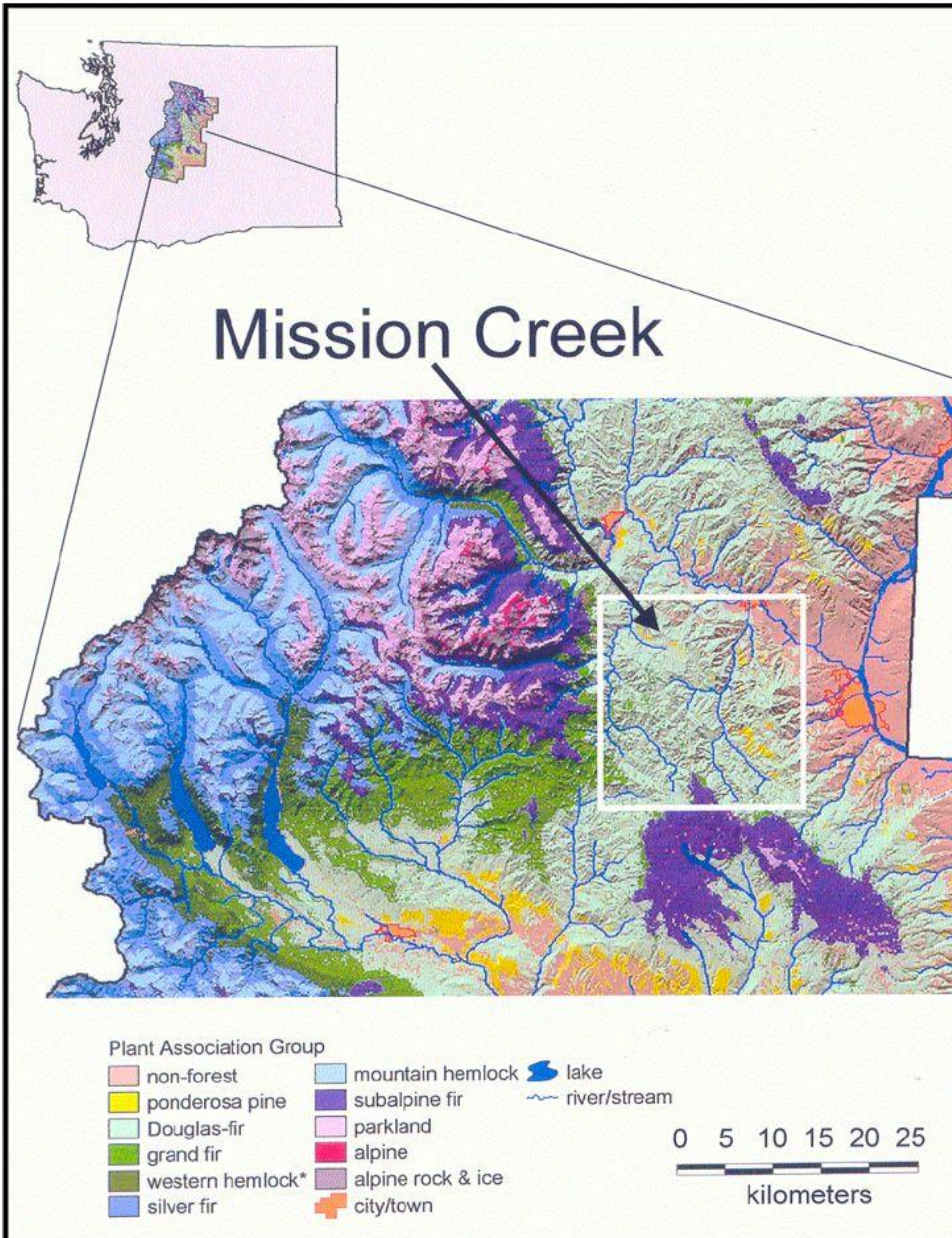
Mission Creek
Plant Association Group
non-forest
ponderosa pine
Douglas-fir
grand fir
western hemlock*
silver fir
mountain hemlock
subalpine fir
parkland
alpine
alpine rock & ice
city/town
lake
river/stream
0 5 10 15 20 25
kilometers

Figure 1

Location of Mission Creek Fire and Fire Surrogate Study site in the central eastern Cascades of Washington State. The site is in the heart of the dry forest types (Douglas-fir plant associations) of the eastern Cascades. Figure adapted from a graphic supplied by P. Hessburg and B. Salter, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Lab, Wenatchee, WA.

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Figure 2

Aerial view of a thin unit (Crow 6) before treatment. Note clumpiness of stand and substantial amount of open area. Unit averages for canopy parameters and fuel models mask the variability at more local scale.

The forests of Mission Creek, like most forests with a major component of ponderosa pine (Hessburg and Agee 2003), were heavily grazed by livestock from the late 19th century into the 1950s (Holstine 1992), and selective logging of the largest and best form ponderosa pines occurred beginning in the 1890s and continued into the latter half of the 20th century. Fire suppression was effective enough to essentially remove fire as an ecological agent in these drier forests (Everett et al. 2000, Wright and Agee 2004). Recently very large and severe wildland fires have occurred in dry forests adjacent to Mission Creek, including the 1994 Rat Creek fire to the west and the 2004 Fischer Fire to the north. Stands are typically denser than historically and with substantially more basal area (20 m^2 ha^{-1} or more).

Methods

Study Design

General methods for all Fire and Fire Surrogate study sites were developed for each site. The design included 12 experimental units, including three control units, three burn units, three fire surrogate units (thinning, mowing, etc.), and three units where fire and the surrogate treatment both occurred. Units were specified as roughly 10 ha to accommodate breeding bird surveys. At Mission Creek, 12 of a possible 30 potential units were randomly chosen as experimental units, on which one of four treatments was to be applied with three replications: a control, a low thinning (the "fire surrogate" treatment), a burn, and a thin/burn treatment. Treatments were randomly assigned but one burn unit was switched with a control unit due to the logistics of fire control. Thinning was completed in Spring 2003, and burning was scheduled for Spring 2004. However, due to a warm late winter in 2004, spring greenup on units occurred earlier than usual and the wildfire season commenced earlier than usual as well. Only 4 of the 6 burn treatments (2 burn and 2 thin/burn) were accomplished, so the design for the first post-treatment measurement period is unbalanced: 4 control, 4 thin, 2 burn, and 2 thin/burn. Statistical analyses use this unbalanced design (in contrast to the originally planned balanced design with three replications per treatment). The 2 units left unburned were burned in Spring 2006 to rebalance the design for longer-term study, but results are expressed here on the design in place up to 2004.

Unit Treatment

The objective of the thinning was to reduce basal area to 10–14 m^2 ha^{-1} in a non-uniform pattern to mimic natural stand patterns and increase resistance to bark beetle attack (Harrod et al. 1999). Trees were removed in a low thinning, concentrating on smaller commercial tree sizes. Yarding was done by helicopter, so that branches and tops were left on site. Thinned units were slashed (small understory trees cut mechanically by hand) after harvest to fall smaller, unmerchantable stems. Ignition of burn units was by hand and helicopter in spring. Fuel moistures ranged from 9–10%, 11–13%, and 10.5–12% for 1-hr, 10-hr and 100-hr timelag fuels. Live fuel moistures were not measured, but typical early season herbaceous fuel moisture in this area is 175% (Agee et al. 2002). Air temperatures during the fires ranged from 13–23°C, and relative humidities ranged from 30–42%. Windspeeds varied from 0–16 km hr^{-1}. Flame lengths ranged from 0.2–1.0 m. Because of early greenup in 2004, fire spread was patchy. On each burn unit, post-fire sampling teams visually estimated percent area burned on a 900 m^2 plot surrounding each of 30–35 sampling points per unit, so approximately 30 percent of each unit was sampled. Burn-only units had fire coverage of 35 and 50%, while thin-burn units had coverage of 23 and 51%.

Unit Sampling

Fuels and selected vegetation parameters that are used in fire behavior and effects predictions were measured before and after burning on all units. Each unit was gridded with 40 × 40 m cells, and alternate gridpoints in rows and columns were selected to sample at least 30 gridpoints per experimental unit. Fuels were measured along two 20 m line transects at each gridpoint. Direction for one transect was randomly selected, and the other was selected randomly within the opposite 180° hemisphere of the first. Each transect began 5 m from the gridpoint to avoid surface disruption by the various disciplinary teams using the unit. Standard line transects for dead and down fuels were applied (Brown 1974), with 1-hr timelag fuels (0–0.62 cm dia) measured in the first 2 m, 10-hr timelag fuels (0.63–2.54 cm dia) from 0–3 m, and 100 hr timelag fuels (2.55–7.62 cm dia) from 0–5 m. Larger fuel timelag classes (1000+-hr timelag) were measured along the entire 20 m transect. On a given unit, then, at least 120 m, 180 m, 300 m, and 1200 m of 1–1000+-hr timelag fuel transects were measured. Fuelbed depth was measured three times along each transect, and litter and duff depth was also measured three times. After treatment, all transects were remeasured using the same methods.

Vegetation variables representing those parameters useful in fire behavior prediction (Scott and Reinhardt 2001a) were measured at each sample point within an experimental unit. Other vegetation sampling concerning floristics was sampled by another team (R. Harrod, Okanogan-Wenatchee National Forests) and is not included here. We sampled canopy closure, herb and shrub biomass, and tree data sufficient to develop estimates of tree height, canopy base height, canopy bulk density, basal area, and tree list data needed for fire effects simulations (variables listed above plus tree density, diameter by size class, and live crown ratio or live crown base) using equations available in FMA Plus (Carlton 2004).

Canopy closure was measured the same way before and after sampling using a convex spherical densiometer with four readings at 90° increments relative to aspect at each sample point. Herb and shrub biomass were measured using the system of Burgan and Rothermel (1984). These vegetation parameters were ocularly estimated over a 400 m^2 area at each sample point into type class (fine or coarse), density class, cover class, and depth, using a photo guide. Tree data for the fire portion of the study were collected using different methods before and after treatment. Before treatment, variable area plots were designed to yield at least 10 trees per sample, and plots ranged from 100–400 m^2. After treatment, due to potential sampling bias associated with variable area plots that might inflate tree density and basal areas, data were collected using variable radius plots with a 10 or 20 factor (English unit) prism. Individual tree measurements included species, diameter at breast height, live crown base and total height.

Analytical Methods

Fuel data for 1- to 1000+-hr timelag fuels and fuelbed depth were compiled using the methods described in Brown (1974). Total forest floor mass (n = 108) was calculated from site-specific equations relating depth to mass (r^2 = 0.84) developed from samples collected in the Mission Creek area (Lolley 2005). Sample mass was adjusted for mineral content before the regressions were run.

Pre- and post-treatment shrub and herb data were exported to FUELCALC (Scott and Lansing 2001). Tree list data were used to calculate canopy base height, canopy bulk density, and basal area using the procedures in FVS-FFE (Reinhardt and Crookston 2003, Carlton 2004). Canopy base height is defined here as the lowest height above which a minimum of 0.011 kg m^{-3} of canopy fuels is present, and canopy bulk density is defined as the maximum 5 m running mean of canopy bulk density calculated in 0.3 m increments through the canopy.

Because of the unbalanced design (4 control and thin units, and 2 burn and thin-burn units) all analyses employed marginal sums of squares, also called type III sums of squares. This results in a more conservative test (Zar 1999). A Type I error of 0.05 was used as the criterion for a significant difference, and all differences reported, listed with their appropriate P-values, met that criterion.

A standard 2-way analysis of variance (ANOVA) design was applied to pre-treatment vegetation and fuel data, with thin and burn as factors (although treatment had not yet occurred). Treatment effects (pre-treatment minus post-treatment) were analyzed using different techniques for vegetation and fuel characteristics. Those vegetation variables where slightly different methods were employed in the pre- and post-treatment vegetation sampling (herb biomass, shrub biomass, canopy base height, canopy bulk density, and basal area) were analyzed with a 2-way analysis of covariance (ANCOVA), with the pre-treatment value used as the covariate. Differences in canopy cover and all fuel characteristics were analyzed using a 2-way ANOVA. All analyses met normality assumptions as analyzed by rankit plots of standardized residuals. Although ANOVA is relatively robust against heterogeneity of variance, it was analyzed for each variable by comparing the standardized residuals with fitted values, and where needed, a transformation to natural logarithm or square root was attempted. For pre-treatment variables, herb biomass, 1-hr timelag, and 1000+-hr rotten were identified as needing a transformation, but transformations did not help. For differences due to treatment, herb biomass, canopy base height, and forest floor biomass were identified as needing a transformation, and transformations were not useful in correcting the problem. The results for those variables, if significant, should be interpreted cautiously.

Potential fire behavior due to treatment was analyzed using post-treatment fuel conditions under 80 and 97 percentile fire weather. A custom fuel model was built for each unit for the four post-treatment conditions (control, thin, burn, and thin/burn) using actual fuel loading measured, with equivalent mass of 0.5 cm depth of the forest floor added to the 1-hr load for non-burn units and a proportional addition on burn plots as a function of percent area burned on each experimental unit. This addition was justified because the surface needles of the forest floor do contribute to the leading edge of surface fire behavior but are not tallied in the line intersect transects. Although in actual field use, considerable alteration of custom fuel models is expected, for the model comparison in this paper we were evaluating relative differences in treatment, so we used the actual measured values on each unit to construct the fuel model. The only way to truly evaluate the model would be to test it under 80 and 97 percentile weather, and there would be few areas well enough contained to allow ignition of such fires. Fire behavior was estimated using NEXUS (Scott and Reinhardt 2001b) with 80 and 97 percentile values taken from two nearby Remote Automated Weather Stations. The 80 percentile values used were: 1-hr TL, 4%; 10-hr TL, 5%; 100-hr TL, 8%; herb, 32%; live woody moisture, 100%; and 6.1 m open windspeed of 19 km hr^{-1}. The 97 percentile values were: 1-hr TL, 3%; 10-hr TL, 4%; 100-hr TL, 6%; herb, 31%; live woody moisture, 90%; and 6.1 m open windspeed of 36 km hr^{-1}. Windspeeds were adjusted based on canopy closure for each unit using factors in Reinhardt and Crookston (2003). Flame lengths, torching index (open windspeed needed to initiate crown fire), and crowning index (open windspeed needed to maintain active crown fire) were recorded from the NEXUS output for both 80 and 97 percentile weather. As each unit had a unique slope that could have confounded response to treatment, fire behavior was estimated on the basis of actual unit slope as well as a fixed 45% slope for all units, which was the grand mean of slope across all units.

The minimum level of treatment effectiveness required by the National Fire and Fire Surrogate network is 80% of the basal area surviving under 80 percentile fire weather, but most large wildfires occur under more extreme fire weather. Potential flame lengths from NEXUS were applied to the post-treatment tree lists using both the 80 and 97 percentile weather in FOFEM (Reinhardt et al. 1997) after binning the trees into 7.6 cm diameter classes. Short-term success of the treatment was determined by units exceeding 80% basal area survival under both sets of weather conditions.

A standard two-way ANOVA design was applied to analyze post-treatment values for 80 and 97 percentile weather flame length, torching index, crowning index, and percent basal area survival. Percent survival was transformed with an arcsine to preserve normality assumptions. All variables were normally distributed and met assumptions of homoscedasticity. The use of unit averages does mask some potential for torching at a more local scale (sample point area of ~400 m^2 compared to unit sizes of 10 ha), so the torching results are also expressed at this scale.

Results

Pre-Treatment Vegetation and Fuels

There were few differences in pre-treatment vegetation parameters when grouped by proposed treatment (Lolley 2005). The only difference was that herbaceous biomass was less ($P = 0.042$) on plots scheduled for burn treatments (Table 1). Fuel differences were minor as well (Table 2), although again individual experimental units did vary considerably. The only difference ($P = 0.048$) was the interaction term for 1000+-hr rotten fuels, suggesting more fuel on plots scheduled for burning than not burned, while there was less 1000+-hr rotten load on plots scheduled for thinning than those not to be thinned.

Table 1

Vegetation characteristics before treatment. Canopy base height, canopy bulk density, and basal area are not directly comparable to posttreatment data because of slightly different methods of measurement (see text). Values in parentheses are standard errors.

Treatment	Herb Biomass (Mg ha^{-1})	Shrub Biomass (Mg ha^{-1})	Canopy Closure (%)	Canopy Base Height (m)	Canopy Bulk Density (kg m^{-3})	Basal Area (m^2 ha^{-1})
Control	0.49 (.05)	4.1 (1.1)	69 (2)	2.8 (.1)	0.061 (.004)	26.2 (2.0)
Thin	0.76 (.23)	4.4 (0.7)	71 (4)	2.9 (.9)	0.063 (.018)	24.0 (1.9)
Burn	0.21 (.08)	7.3 (1.5)	69 (5)	1.8 (.6)	0.056 (.004)	24.4 (0)1
Thin/Burn	0.19 (.09)	4.2 (1.6)	69 (4)	1.4 (.2)	0.069 (.013)	27.2 (1.3)

Table 2

Fuel characteristics before treatment. Values in parentheses are standard errors.

Treatment	1-hr Load (Mg ha⁻¹)	10-hr Load (Mg ha⁻¹)	100-hr Load (Mg ha⁻¹)	1000+-hr Sound Load (Mg ha⁻¹)	1000+-hr Rotten Load (Mg ha⁻¹)	Forest Floor Depth (cm)	Forest Floor Mass (Mg ha⁻¹)	Fuelbed Depth (cm)
Control	1.07 (.34)	2.01 (.26)	3.52 (.70)	8.3 (2.9)	3.1 (.6)	5.04 (5.3)	38.9 (4.2)	9.8 (1.2)
Thin	0.83 (.31)	1.23 (.14)	5.22 (.56)	11.5 (2.1)	9.4 (2.8)	4.57 (6.1)	35.2 (4.8)	10.4 (2.0)
Burn	1.42 (.06)	2.01 (.32)	4.94 (1.9)	8.7 (3.6)	9.7 (.1)	4.80 (1.6)	37.0 (1.3)	9.7 (0.1)
Thin/Burn	1.80 (.08)	2.29 (.02)	5.69 (.74)	14.6 (3.1)	6.7 (2.2)	3.60 (7.0)	27.5 (5.6)	14.5 (3.3)

Treatment Effects on Vegetation and Fuels

Thinning and burning had effects on both vegetation and fuel parameters (Tables 3 and 4, Figure 3). Thinning tended to decrease the vegetation variables and increase surface fuel loading, while burning tended to have little effect on the selected vegetation variables and decreased most surface fuels. Thinning had effects on canopy variables: canopy closure decreased ($P = 0.013$), canopy base height increased ($P = 0.006$), canopy bulk density decreased ($P = 0.0001$), and basal area decreased ($P = 0.008$). Thinning had no short-term influence on herb and shrub biomass. Burning had no influence on any vegetation variable. Its influence on herb biomass was likely muted both by patchy fire coverage of the spring burns and by regrowth between the time of the fire in Spring 2004 and measurement in Summer 2004.

Table 3

Vegetation characteristics after treatment. Canopy base height, canopy bulk density, and basal area are not directly comparable to pretreatment data because of slightly different methods of measurement (see text). Values in parentheses are standard errors.

Treatment	Herb Biomass (Mg ha⁻¹)	Shrub Biomass (Mg ha⁻¹)	Canopy Closure (%)	Canopy Base Height (m)	Canopy Bulk Density (kg m⁻³)	Basal Area (m² ha⁻¹)
Control	0.42 (.09)	1.01 (.05)	72 (2)	6.4 (1.1)	0.050 (.005)	19.1 (0.5)
Thin	0.41 (.17)	1.16 (0.2)	56 (9)	6.8 (0.4)	0.030 (.005)	12.2 (1.4)
Burn	0.15 (.07)	1.74 (0.6)	69 (2)	3.8 (0.4)	0.037 (.003)	15.9 (0.9)
Thin/Burn	0.24 (.13)	1.15 (.02)	57 (6)	5.5 (0.2)	0.033 (.006)	15.5 (1.4)

Table 4
Fuel characteristics after treatment. Values in parentheses are standard errors.

Treatment	1-hr Load (Mg ha⁻¹)	10-hr Load (Mg ha⁻¹)	100-hr Load (Mg ha⁻¹)	1000+-hr Sound Load (Mg ha⁻¹)	1000+-hr Rotten Load (Mg ha⁻¹)	Forest Floor Depth (cm)	Forest Floor Mass (Mg ha⁻¹)	Fuelbed Depth (cm)
Control	0.44 (.16)	1.01 (.10)	2.10 (.34)	8.4 (2.8)	3.4 (.9)	5.78 (.55)	44.7 (4.3)	11.4 (1.8)
Thin	1.14 (.34)	3.57 (.38)	6.29 (.51)	18.8 (1.0)	6.5 (1.1)	3.42 (.31)	48.7 (2.5)	24.5 (3.6)

Treatment	1-hr Load (Mg ha^{-1})	10-hr Load (Mg ha^{-1})	100-hr Load (Mg ha^{-1})	1000+-hr Sound Load (Mg ha^{-1})	1000+-hr Rotten Load (Mg ha^{-1})	Forest Floor Depth (cm)	Forest Floor Mass (Mg ha^{-1})	Fuelbed Depth (cm)
Burn	0.71 (.07)	1.02 (.09)	2.40 (.14)	3.0 (0.7)	3.8 (2.7)	6.27 (.35)	26.1 (2.8)	13.7 (4.9)
Thin/Burn	0.81 (.07)	1.61 (.09)	4.35 (.49)	13.5 (4.7)	2.0 (.7)	3.57 (.71)	27.3 (0.6)	13.4 (1.6)

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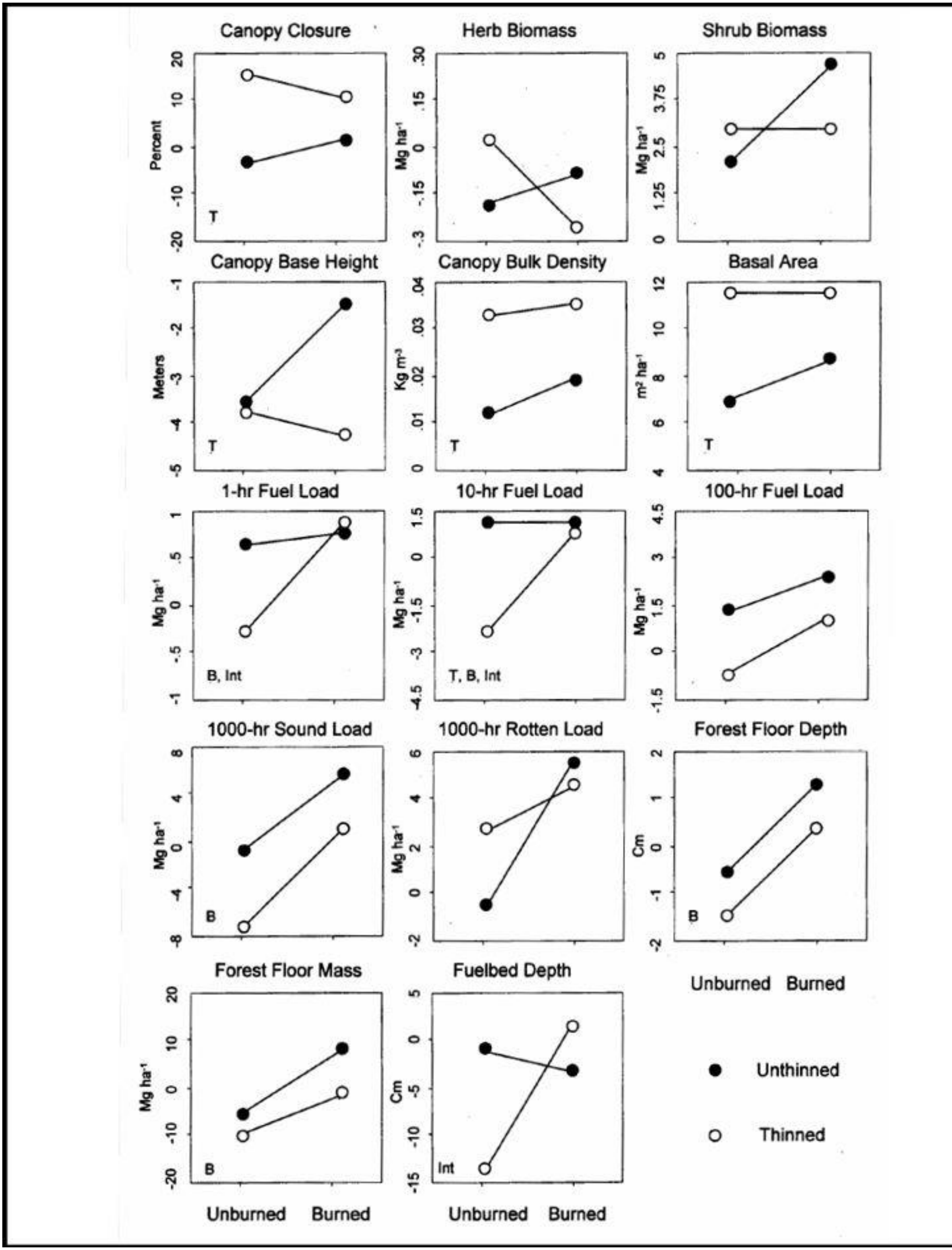
Canopy Closure
Herb Biomass
Shrub Biomass
Canopy Base Height
Canopy Bulk Density
Basal Area
1-hr Fuel Load
10-hr Fuel Load
100-hr Fuel Load
1000-hr Sound Load
1000-hr Rotten Load
Forest Floor Depth
Forest Floor Mass
Fuelbed Depth
Percent
Mg ha-1
Meters
Kg m-3
m2 ha-1
Cm
T
B, Int
T, B, Int
B
Int
Unburned Burned
Unthinned
Thinned

Figure 3

Differences in vegetation and fuel parameters due to treatment. The y-axis on each graph contains the range of differences (before minus after) for each variable. On the left side of each graph are unburned unit averages, and on the right side are burned unit averages. Filled circles are unthinned unit averages, and open circles are thinned unit averages. Significant effects ($P < 0.05$) are listed in the lower left of each graph: B = significant burn effect; T = significant thin effect; Int = significant interaction effect. Note that scales and units differ by graph.

The pre-treatment basal area on control plots, measured with variable area plots, had roughly 7 $m^2 ha^{-1}$ more basal area than the same control plots measured after treatment with variable radius plots, suggesting that a bias did occur from the use of variable area plots, and substantiating the use of pre-treatment values as covariates in the analysis.

Thinning increased ($P = 0.001$) 10-hr TL fuels. Burning decreased 1-hr TL fuels ($P = 0.003$), 10-hr TL fuels ($P = 0.003$), 1000+-hr TL Sound ($P = 0.033$), forest floor depth ($P=0.005$), and forest floor mass ($P = 0.006$). There were interactions between thinning and burning for 1-hr TL fuels ($P = 0.006$), 10-hr TL fuels ($P = 0.003$), and fuelbed depth. The fuel mass interactions on thinned units occurred because of substantial fuel consumption on thinned units that were burned, likely due to more continuous and higher fuel loads after thinning. The interaction with fuelbed depth is due to large increases on thinned units, compared to almost no increase on those that were thinned and burned.

Potential Fire Behavior and Tree Mortality

No changes in significant differences occurred from the two slope conditions (use of actual unit slope versus grand mean of all unit slopes) so results are presented using the actual slope values. Potential flame lengths predicted after treatment under 80 percentile weather did not differ between treatments (Table 5). Thinning increased flame length at 97 percentile fire weather (Table 6). There was no torching potential identified at the unit scale, and there were no differences between treatments. However, within a thin/burn unit, with an average canopy base height of 5.3 m and no torching potential at the unit scale, 15% of the sample plots had a canopy base height less than 2 m, indicating some torching potential at the plot scale under the conditions simulated. Similar variability was found in all units. Active crown fire potential, expressed as crowning index, was significantly lower on the thin treatment (a higher windspeed is necessary).

Table 5

Treatment effects on flame length, torching index, crowning index, and basal area survival with 80-percentile fire weather. Values in parentheses are standard errors.

Treatment	Flame Length (m)	Torching Index (km hr^{-1})	Crowning Index (km hr^{-1})	Basal Area Survival[1] (%)	Residual Basal Area (m^2 ha^{-1})
Control	1.01 (.09)	163 (19)	64 (4)	84 (.85)	17.6 (.66)
Thin	1.26 (.23)	142 (51)	95 (4)	80 (0)	11.2 (1.43)
Burn	0.93 (.06)	138 (34)	83 (8)	83 (2.33)	14.9 (1.55)
Thin/Burn	1.22 (.24)	133 (65)	87 (14)	81 (1.50)	14.6 (2.17)

[1] averages of percentages within units, not across treatment category

Table 6
Treatment effects on torching index, crowning index, and basal area survival with 97 percentile fire weather. Values in parentheses are standard errors.

Treatment	Flame Length (m)	Torching Index (km hr^{-1})	Crowning Index (km hr^{-1})	Basal Area Survival[1] (%)	Residual Basal Area (m^2 ha^{-1})
Control	1.39 (.12)	130 (16)	58 (4)	72 (8.1)	14.9 (1.8)
Thin	1.90 (.34)	114(41)	87 (4)	44 (5.5)	6.7 (0.3)

Treatment	Flame Length (m)	Torching Index (km hr^{-1})	Crowning Index (km hr^{-1})	Basal Area Survival[1] (%)	Residual Basal Area (m^2 ha^{-1})
Burn	1.25 (.10)	107 (28)	75 (8)	75 (11.5)	13.2 (2.4)
Thin/Burn	1.81 (.56)	105 (52)	79 (13)	48 (31.0)	9.2 (6.6)

[1] averages of percentages within units, not across treatment category

Basal area survival expressed as either residual basal area or percent survival did not differ between treatments under 80 or 97 percentile weather. Although the treatment means appear quite different (Tables 5 and 6), and the effect of thinning at 80 percentile (P = 0.055) and 97 percentile (P = 0.078) is close to significant, there is substantial variability among units. Simulated wildfire under less severe weather thins the units from below, and fire under more severe weather removes some larger trees (Figure 4).

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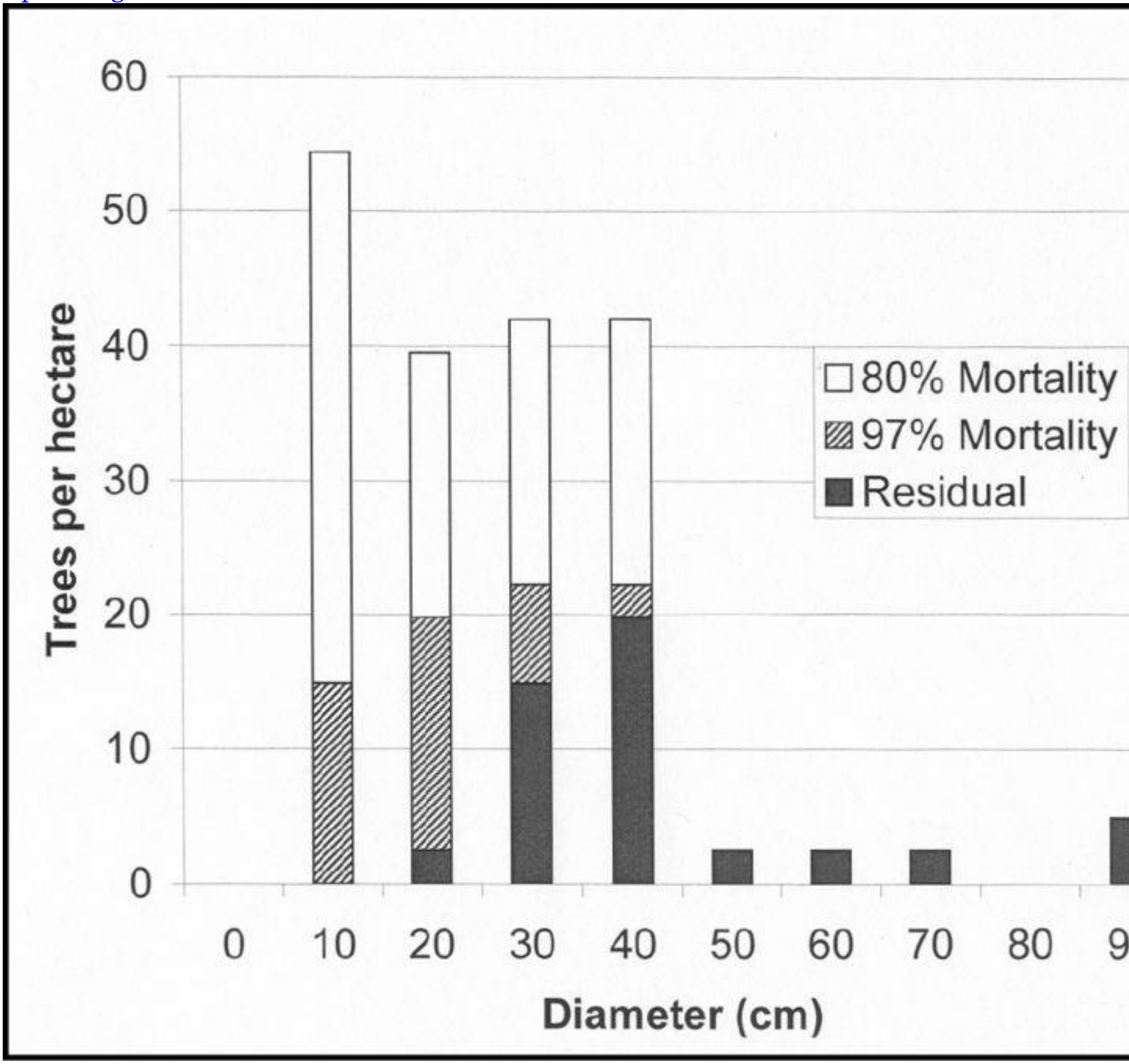

Figure 4

Potential wildfire (80 and 97 percentile weather) effects on stand structure for a control unit. Pre-wildfire density by size class is shown by the total height of each column. Potential mortality under 80-percentile weather is shown as the white portion of each column. Additional mortality under 97 percentile weather is shown by the lined portion of the column, and residual trees under 97 percentile weather are shown as the black portion of the column.

Discussion

Thinning without subsequent slash reduction has been shown in simulation studies to be less effective than prescribed fire in reducing potential wildland fire behavior (van Wagtendonk 1996, Stephens 1998, Agee and Skinner 2005, Stephens and Moghaddas 2005). This study indicated that thinning, without slash reduction because of the helicopter yarding method used, increased surface fire behavior and brought stands close to historic basal area levels. The slash left after logging, plus smaller trees that were not logged, were treated after the thinning with a lop and scatter method. This expensive additional cost might not occur after every thinning operation, but can aid subsequent survival of residual trees (Agee and Skinner 2005). However, addition of dead fuels from thinning or insect outbreaks (Hummel and Agee 2003), or removal by prescribed fire, does not simply translate into altered fire hazard or severity. Fuelbed depth, surface-area-to-volume ratios, and distribution of loading by size class are fuel model parameters in addition to total loading that will affect fire behavior.

Spring burning was not effective in significantly altering stand structure or reducing potential surface wildfire behavior in this study. In the judgment of the management and research personnel on site at the time of the prescribed fires, the fires were ineffective in meeting fuel management objectives because the fire spread was so patchy. Dead fuel moistures were in prescription, but early greenup of herbaceous fuels had a dampening effect on fire spread. Two of the initially planned fires were cancelled, and only two of the other four burns had fire coverage exceeding 50%. So 50–75% of the area in burn treatments in effect received no treatment at all. Total dead and down woody fuel load was decreased 23–55% in burn and thin-burn units. Results from two Fire and Fire Surrogate sites in California mixed conifer forests (Blodgett Forest and Sequoia National Park) showed that the dead and down fuels were reduced 88% (Sequoia) to 90% (Blodgett) by fall burns, while spring burns (Sequoia) consumed 67% of dead and down fuels (Stephens and Moghaddas 2005, Knapp et al. 2005). At Blodgett, mechanical, mechanical plus fire, and fire-only treatments all significantly reduced fire behavior relative to controls; their mechanical treatment included thinning and subsequent mastication (shredding and chipping) of small live and dead trees (Stephens and Moghaddas 2005). Choice of season of burn and thinning technique is clearly important in predicting effects on post-treatment fire behavior.

The minimum standard for treatment effectiveness used by the Fire and Fire Surrogate network is 80% survival under 80 percentile weather conditions. Under these conditions and the stand structures at Mission Creek, there was substantial survival of the basal area of the stands regardless of treatment (Table 5), so that all treatments expressed by averages at the unit scale (including the control units), met the 80–80 criterion for survival. The projected residual basal area after the simulated 80 percentile weather wildfire ranged from 11–17 m^2 ha^{-1}. The moderate wildfire flame lengths (0.9–1.2 m) and large tree sizes left even on thinned plots allowed most of the large trees to survive under the 80 percentile fire weather. More typically for hazard reduction planning, wildfire scenarios are run under more extreme fire weather conditions (90–97 percentile fire weather), and at Mission Creek under these more extreme conditions, tree mortality did not differ between treatments, although mortality exceeded that under 80 percentile fire weather. As discussed below, interpretation of these effects is partly a function of scale.

Most fire effects simulations rely on first-order effects of fire: the projected effects of heating the cambium and scorching the crown. They do not take into account second-order effects, either from synergistic effects with other processes (e.g., insects) or long-term smoldering (Swezy and Agee 1991, Perrakis 2005). The post-treatment fuel inventories show that there is much more fuel on the thinned plots than burned plots, and in addition to increased surface fire flame lengths under extreme fire weather, thinned units would likely be subject to longer smoldering periods and more damage to trees, understory vegetation, and possibly soils. The 1000+-hr fuels plus forest floor mass total over 60 Mg ha^{-1} on thin-only units compared to 35 Mg ha^{-1} on thin-burn units and less than 25 Mg ha^{-1} on burn-only units. In a subsequent wildfire, much of that biomass would be consumed, and the surface heating might cause additional tree mortality.

Torching potential expressed at the unit scale was low in this study, but the bias in the measurements before and after treatment (variable area versus variable radius plots to inventory trees) makes direct comparison of pre- and post-treatment data risky. We believe the variable radius plots used after treatment were more reliable, because prism sampling is unbiased. Occasional plots contained no trees. In contrast, the variable area plots were expanded as needed to include at least 10 trees per sample, so that every plot contained trees, and this resulted in higher basal area expressed at the unit scale (see control unit basal areas in Tables 1 and 3). Comparison of torching index values for just the post-treatment data show little torching potential.

The low torching potential is contradictory to local and regional experience on recent wildfires, where torching potential appears to be substantial (Williamson 1999, R. Harrod and K. Satterfield, Okanogan-Wenatchee National Forests, personal communication). Resolution of this apparent contradiction lies in the scale at which the data are presented. Unit averages were comprised of about 30 plot measurements each, and across all units, 17% of these plots had sufficiently high surface fireline intensity and sufficiently low canopy base height values such that torching would be enabled with 97 percentile weather. As not only the torched plot but surrounding trees would also be killed from the heat generated by the torched group, perhaps a third of the stand would be killed simply from the torching, above any mortality from surface fire activity alone. This is likely the scale at which the forest would actually experience the fire, and it is consistent with local wildfire experience. However, the experimental design, if analyzed at the plot scale, would have been pseudoreplicated (Hurlbert 1984), so that statistical analysis at the plot scale is not possible.

Crown fire hazard is often cited as a justification for stand treatment, particularly the need to thin in dry forests. Yet until recently, quantitative analysis of crown fire hazard was difficult. Models that link surface and crown fire behavior are now being applied widely (Scott and Reinhardt 2001a). Using variants of earlier models (e.g., Van Wagner 1977, Rothermel 1991), assessments of crown fire risk can be produced that are based either on stand characteristics (canopy bulk density, canopy base height) or weather conditions needed to initiate or sustain crown fire behavior (torching index and crowning index, both defined in windspeed units). Although the models are empirical, they provide a standard for comparison of risk. In this study, there was little risk of active crown fire before or after treatment with canopy bulk density values below 0.070 kg m^{-3} (Tables 1 and 3) and crowning indices well above expected windspeeds in the area (Tables 5 and 6). Thinning increased crowning index and therefore reduced crown fire hazard. But whatever the crowning index, it must be compared to expected windspeeds in the area in order to identify real risk. Arbitrary thresholds that define low, moderate, and high crowning potential across a state or region (e.g., Fiedler et al. 2004) need to have a more site-specific interpretation based on local wind data. Given our worst case windspeeds of 36 km hr^{-1}, and the crowning index average across all units under 97 percentile weather (Table 6), thinning to reduce active crown fire potential is an unnecessary action for these conditions. However, thinning may be effective in reducing torching potential, and the potential for bark beetle attack, which would have its own fuel accretion implications over time.

The Fire and Fire Surrogates national network is anticipated to continue over periods of times long enough to repeat treatments, such as prescribed fire in the short-term (5–10 years) and repeat thinning over longer time periods (20–30 years). This will allow quantification of the effect of maintenance treatments over time, and the longer-term efficacy of prescribed fire and thinning on forest health.

Notes

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